Fish Assemblages in the Upper Esopus Creek, NY: Current Status, Variability, and Controlling Factors

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Barry P. Baldigo[1,*], Scott D. George[1], and Walter T. Keller[2]

Abstract - The Upper Esopus Creek receives water diversions from a neighboring basin through the Shandaken Tunnel (the portal) from the Schoharie Reservoir. Although the portal is closed during floods, mean flows and turbidity of portal waters are generally greater than in Esopus Creek above their confluence. These conditions could potentially affect local fish assemblages, yet such effects have not been assessed in this highly regulated stream. We studied water quality, hydrology, temperature, and fish assemblages at 18 sites in the Upper Esopus Creek during 2009–2011 to characterize the effects of the portal input on resident-fish assemblages and to document the status of the fishery resource. In general, fish-community richness increased by 2–3 species at mainstem sites near the portal, and median density and biomass of fish communities at sites downstream of the portal were significantly lower than they were at sites upstream of the portal. Median densities of *Salmo trutta* (Brown Trout) and all trout species were significantly lower than at mainstem sites downstream from the portal—25.1 fish/0.1 ha and 148.9 fish/0.1 ha, respectively—than at mainstem sites upstream from the portal—68.8 fish/0.1 ha and 357.7 fish/0.1 ha, respectively—yet median biomass for Brown Trout and all trout did not differ between sites from both reaches. The median density of young-of-year Brown Trout at downstream sites (9.3 fish/0.1 ha) was significantly lower than at upstream sites (33.9 fish/0.1 ha). Waters from the portal appeared to adversely affect the density and biomass of young-of-year Brown Trout, but lower temperatures and increased flows also improved habitat quality for mature trout at downstream sites during summer. These findings, and those from companion studies, indicate that moderately turbid waters from the portal had few if any adverse impacts on trout populations and overall fish communities in the Upper Esopus Creek during this study.

Introduction

The Upper Esopus Creek is a historic trout fishery and recreational stream in the Catskill Mountains of southeastern New York state. The Shandaken Tunnel (the portal) delivers water that is usually cool during the summer from the Schoharie Reservoir to the Upper Esopus Creek (hereafter, the Esopus), and maintains a minimum flow in the Esopus as stipulated by state regulations (CCE 2007). On average, these releases are more turbid than the waters of the Esopus and have been the focus of controversy because of the perception that the turbidity of the portal's inflow waters negatively affects water quality and trout populations in the upper basin (CCE 2007). Any stressors that adversely affect *Salmo trutta* L. (Brown Trout) and *Oncorhynchus mykiss* (Walbaum) (Rainbow Trout) populations might negatively

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affect the local economy because these fisheries are important resources. Both the concentration of suspended sediment and the duration of turbid episodes can adversely affect the health of lotic species and thus aquatic communities (Newcombe and Macdonald 1991, Shaw and Richardson 2001). Turbidity has been linked with decreased visual acuity and growth rates in trout under laboratory conditions (Shaw and Richardson 2001; Sigler et al. 1984; Sweka and Hartman 2001a, b), reduced density and biomass of macroinvertebrates in the wild (Wagener and LaPerriere 1985), reduced biomass and primary productivity in periphyton (Bilotta and Brazier 2008, Quinn et al. 1992), and siltation-related habitat degradation in all 3 communities (Henley et al. 2000). Field and mesocosm studies, moreover, have found a range of neutral to negative impacts of turbidity on feeding, growth, density, and biomass of resident trout or their populations (Redding 1987, Sweka and Hartman 2001a, White and Harvey 2007). Because there are few data on fish communities and turbidity in the Esopus, the potential effects of turbidity on local fish communities and trout populations in this basin are largely speculative and subject to debate.

Discharge of relatively cool water from the portal could also have beneficial effects on cold-water species. Water temperatures in parts of the Esopus frequently exceed upper thermal limits for growth (~19 °C) and survival (~25 °C) of Brown Trout (Hasnain et al. 2013, Wehrly et al. 2007) for long low-flow periods that occur during most summers (Ross 2012). The portal provides supplemental flows that can greatly exceed natural flows in the mainstem Esopus (at their juncture) by as much as 1 order of magnitude. The temperature of portal water often is below the upper lethal limits for Brown Trout during warm months, and may allow the species to survive, if not grow, in nearby reaches immediately downstream from the portal that otherwise might be uninhabitable.

Several factors other than the portal input may affect water quality and impact stream ecosystems in the Upper Esopus Basin including (1) effluent discharges from sewage-treatment plants within the watershed; (2) runoff from a ski area; (3) withdrawals from streams for various purposes, including snow-making, water supply, and bottling; and (4) areas with high concentrations of septic-systems adjacent to tributaries. These factors, combined with water releases from the portal, subject local fish populations and communities to a range of thermal, hydrologic, and waste-water stresses in the upper basin. Unbiased contemporary data are needed to quantify the status of water quality, fish assemblages, and potential impairments in the Esopus.

In 2009, the US Geological Survey (USGS), in collaboration with the New York State Department of Environmental Conservation (NYSDEC), New York City Department of Environmental Protection (NYCDEP), and Cornell Cooperative Extension of Ulster County (CCEUC), began a comprehensive assessment of the biological condition (fish, invertebrates, and diatoms) and water quality (turbidity and nutrients) in the Esopus. Primary objectives of this study were to assemble contemporary data to define the effects of discharges from the portal on resident fish assemblages and to foster better decision-making regarding the management of the fishery, water quality, and water quantity (allocated water from the Schoharie Reservoir to the Ashokan Reservoir) in the Esopus. Specific goals of this effort were

to characterize the natural variability and its relationship with water quality for the biological communities, and to assess the potential effects of point sources of turbidity on fish assemblages. We gave special attention to trout populations because of their economic value and location at the top of the aquatic food chain, as well as the perception that the local fishery and other natural resources within the basin have declined due to the turbid waters delivered to the upper basin via the portal. In this paper, we describe the status of resident fish assemblages and trout populations at 18 sites in the Esopus that we sampled annually, with a few exceptions at selected sites, from June through August during 2009–2011.

Study Area

The Upper Esopus Creek Basin is located in the south-central Catskill Mountains of southeastern New York and follows a 41.8-km semi-circular course from its headwaters at Winnisook Lake, to its terminus at the Ashokan Reservoir (Fig. 1). Its 497-km^2 watershed is contained entirely within the Catskill Park and drains some of the most rugged and mountainous terrain in the Park. Forested land comprises over 95% of the watershed, and commercial and residential development occupies the rest (CCE 2007). The Shandaken Portal and the Esopus confluence occurs in the village of Shandaken just west of Phoenicia (Fig. 1). As of about 2005, water discharged from the portal had the highest median turbidity (8.8 Nephelometric Turbidity Units [NTU]) of any tributary previously assessed in the upper basin; however, its contribution to the total annual sediment load of the Esopus was not fully defined (CCE 2007). Channels for several Esopus tributaries cut into clay-rich till and also contrib-
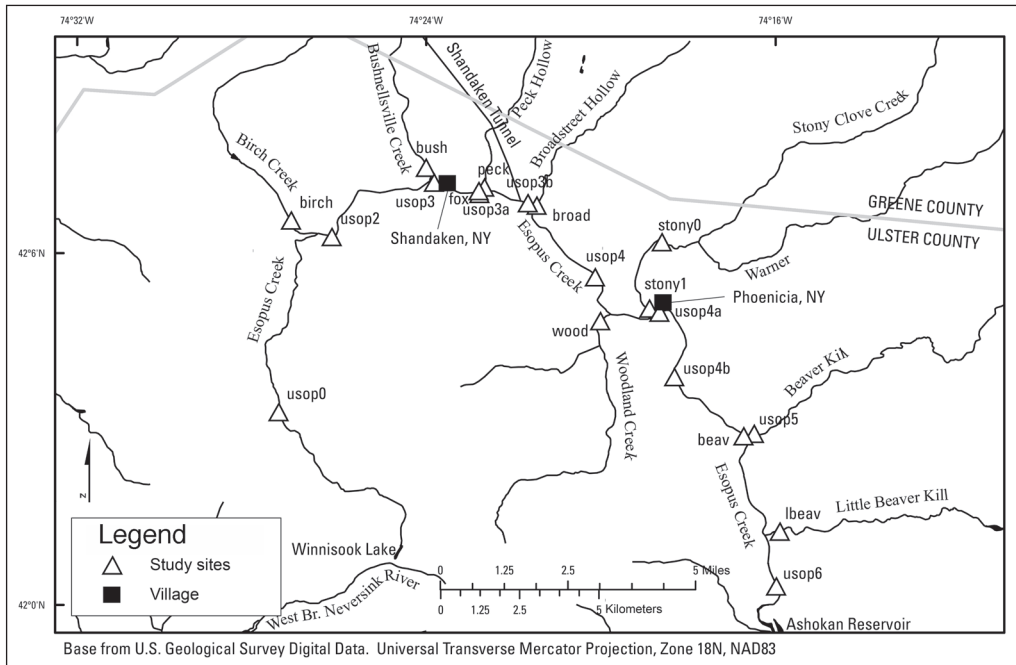

Figure 1. Location of 20 sites on the Upper Esopus Creek surveyed 2009–2011.

ute a significant portion of the total annual sediment load during rain and snow-melt events (CCE 2007). The portal can discharge more than 2 million m^3 of water per day into the Esopus when at maximum output. Since 1977, water releases have been regulated by Part 670 of Title 6 of New York Compilation of Codes, Rules and Regulations (NYCRR; CCE 2007). In 2006, a NY State Pollutant Discharge Elimination System (SPDES) permit placed additional limits on flow, turbidity, and temperature in order to protect the health of aquatic biota in the Esopus (CCE 2007).

Methods

We collected fisheries and water-quality data at 18 sites in 2009 and 2010 and at 16 sites in 2011 (Fig. 1). Of the 18 sites, 10 were on tributaries with drainage areas of 10.3–83.9 km^2, 1 (esop0) is a small headwater mainstem site of 30.3 km^2, and 7 have drainages larger than 100 km^2 and are located on the mainstem of the Upper Esopus (Table 1). Elevations of sites range from 189 m to 455 m (Table 1). At the time, there were no natural or man-made barriers to fish passage in the study area. We distributed mainstem sites evenly above and below the portal in an attempt to distinguish differences in fish communities caused by the portal from those that occurred naturally due to normal shifts in stream conditions between successively larger stream sites.

In general, we chose study sites (reaches) so that they encompassed 2 or more complete geomorphic channel-unit sequences (Fitzpatrick et al. 1998, Meador et al. 2003, Simonson et al. 1994). Most study reaches were 20–35 mean-channel widths long (maximum of 100 m). At each reach, we collected fish by electrofishing up to a 100-m-long section using a backpack shocker (Smith-Root Model 12B) and 3 or 4 netters. After blocking each reach with seines, quantitative surveys consisted of 3 electrofishing passes through each reach; sample-area dimensions were measured on site. At sites with narrow channels (less than 15 m), we placed the blocking seines completely across the channel at the upstream and downstream ends of the study section. At sites with wider channels, we completed 3 replicate surveys in relatively small near-shore subreaches. At each subreach, we affixed 1 blocking seine to the bank, stretched it perpendicular to the bank, and attached it to a rock or rebar 6–8 m from shore. We placed a second 25-m seine oriented downstream and parallel to shore, attached it to a second rock or rebar, and placed a third seine between the second rebar and shore. Hence, each subreach was 6–8 m wide x 25 m long and was blocked by seines on 3 sides and by shoreline on the fourth. We processed fish collected from each sub-reach and pass separately. We identified each fish to species and recorded the lengths and weights of all individuals longer than 150 mm. For very abundant species (some minnows), which were usually shorter than 150 mm, we measured subsamples and obtained lengths and weights from 40–50 individuals; thereafter, we recorded total weights and counts by species in batches of 10–50. We returned all fish to the stream after processing.

We characterized several key habitat features within each study reach and subreach (see Table A in Supplemental File 1, available online at http://www.eaglehill. us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers,

at http://dx.doi.org/10.1656/N1280.s1). We recorded total reach length and 10 measurements of reach width, and employed a modified point and transect method (Fitzpatrick et al. 1998) to measure depth and velocity and to estimate dominant substrate-size categories at 3 points—center, 25%, and 75% of crossection—along each of 10 evenly spaced transects. We used the total length and mean width to calculate sample areas, and included all measurements in mean depth and velocity calculations. We determined the dominant and subdominant particle sizes from the frequency of each category. Data for mean and median water-year (1 October–30 September) temperature, discharge, suspended-sediment concentration, and turbidity, along with annual suspended-sediment loads at all study sites, were from McHale and Siemion (2014) and USGS (2014).

We determined standard metrics for entire fish communities and selected species populations for each survey (each site and year that they were sampled). We used the number of fish captured during each pass to estimate density and biomass (and 95% confidence intervals [CI]) for the entire fish community and for each species

Table 1. Site names, codes, and USGS site numbers for sites surveyed in the Upper Esopus Creek, 2009–2011. Elev. = elevation.

Stream and site name	Site code	USGS site number	Latitude (°N)	Longitude (°W)	Drainage area (km²)	Elev. (m)
Tributary sites						
Fox Hollow	fox	01362199	42.1161111	74.380556	10.3	309
Peck Hollow	peck	01362215	42.1255556	74.376389	12.3	351
Broadstreet Hollow	broad	01362232	42.1125556	74.358694	23.7	296
Bushnellsville Creek	bush	01362197	42.1247222	74.401139	29.5	336
Esopus Creek at Oliverea[A]	esop0	0136219203	42.0525000	74.456222	30.3	455
Birch Creek	birch	013621955	42.1089787	74.451818	32.4	377
Little Beaverkill	lbeav	01362497	42.0195364	74.266258	42.7	205
Woodland Valley Creek	wood	0136230002	42.0797222	74.334583	53.4	268
Beaverkill	beav	01362487	42.0467580	74.276814	64.7	213
Stony Clove Creek at Chichester	stoc0	01362370	42.1020278	74.310889	80.0	292
Stony Clove Creek at Phoenicia	stoc1	01362398	42.0830556	74.315833	83.9	245
Upstream mainstem sites						
Esopus Creek at Big Indian	esop2	0136219565	42.1041667	74.435833	111.9	355
Esopus Creek at Shandaken	esop3	0136219710	42.1194444	74.397500	152.0	317
Esopus Creek at Allaben	esop3a	01362200	42.1170341	74.380149	165.0	305
Downstream mainstem sites						
Esopus Creek downstream of portal	esop3b	0136223005	42.1133333	74.361889	181.0	287
Esopus Creek upstream of Phoenicia	esop4	01362250	42.0925000	74.335972	215.7	268
Esopus Creek at Phoenicia	esop4a	01362405	42.0819444	74.312028	357.4	238
Esopus Creek downstream of Phoenicia	esop4b	01362420	42.0636111	74.306389	365.2	225
Esopus Creek at Mt Tremper	esop5	01362430	42.0468889	74.280000	373.0	207
Esopus Creek at Boiceville	esop6	01362500	42.0142588	74.270425	497.3	189

[A]Site esop0 was also treated as an upstream mainstem site for selected analyses.

population with the Moran-Zippin method of proportional reduction (Van Deventer and Platts 1985, Zippin 1958). These values were divided by the total area sampled at each site to estimate the number or the biomass (g) of fish in the local community or species population per unit area. Two components of ecosystem diversity, breadth or size (total species richness) and heterogeneity (Simpson's diversity index), were also generated for each survey (Simpson 1949, Whittaker 1975). Richness (*S*) is the number of different fish species collected in the sampled area at each site. Simpson's diversity index is commonly used to characterize community biodiversity, and it employs the number of species present and the relative abundance of each species to calculate a metric that ranges from 0 to 1. When calculated as $1 - D$, zero indicates no diversity (e.g., 0–1 species), and values close to 1 indicate a large number of species and proportionally similar numbers of each.

We described the status of resident-fish populations and communities, and spatial and temporal variations in key metrics graphically. We evaluated the potential effects of the portal on fish assemblages with 4 related methods. Longitudinal and annual trends in metrics (total community density, biomass, richness, and diversity, as well as density and biomass for trout populations) were evaluated through graphical analyses. The significance of changes or differences in metrics between or among individual sites and groups of sites within (a) tributaries, (b) upstream of the portal, and (c) downstream of the portal were evaluated using the upper and lower 95% CIs (which equal 1.96 times the standard errors), and parametric analysis of variance (ANOVA) if variance and normality assumptions were met, or non-parametric Kruskal-Wallis tests if data were not normally distributed. We considered differences in means and medians to be significant when the CIs did not overlap (Cumming et al. 2007) or when *P*-values for appropriate statistical tests were ≤0.05. We completed additional analyses of spatial patterns in fish-community composition and classifications (groupings of sites with similar assemblages) through non-metric multidimensional scaling (MDS) ordination of taxon-density data that were square-root transformed (Kruskal 1964, Shepard 1962). MDS ordination generates an arrangement of samples in "species space" according to the non-parametric ranks of their Bray-Curtis similarities. We employed a similarity profile test (SIMPROF) to identify statistically significant (a priori unstructured) clusters within the larger dendrogram (Clarke and Warwick 2001). The significant groups were labeled as such on a non-metric MDS ordination to visually express the relationships among dissimilar groups. Subsequently, we used a similarity-percentage (SIMPER) analysis to determine the contribution of individual species to the overall dissimilarity between groups of samples (Clarke and Warwick 2001). The SIMPER breaks down the original Bray-Curtis similarities between samples into percent contributions of each species to those sample similarities, and identifies the species that are most responsible for defining site and group dissimilarities.

Results

Site names and the codes we used for our analyses and to report our results are provided in Table 1. Study sites on the mainstem of the Esopus upstream from the

portal are collectively referred to as upstream sites and those downstream from the portal are referred to as downstream sites in the remainder of this manuscript.

Fish communities

Species richness ranged from as low as 4 species at 3 of the small drainage-area (DA) tributaries (fox, peck, and bush) during 1 or 2 years, to a maximum of 15 species at the largest DA site (esop6) during 2009 and 2010 (Tables 1, 2; Fig. 2A). Total community richness was strongly related to the drainage area of study sites (Fig. 3). Mean richness at all tributary sites (DA < 100 km^2) over all years was 7.0 species, whereas mean richness at mainstem sites (DA > 100 km^2) averaged 9.2 species upstream from the portal and 11.1 species downstream from the portal (Table 3). Median richness differed significantly among tributary, upstream, and downstream sites (Table 3). With one exception (birch during 2010), richness at the 6 tributary sites with DAs <40 km^2 ranged from 4 to 7 species. Richness reached an asymptote at about 10 species as DA increased above 40 km^2. Influx of nonresident species from Schoharie Reservoir through the portal may have increased richness at 2 sites closest to the portal. Richness at esop6 was larger than anticipated and inconsistent with the asymptotic pattern—a finding probably related to the influx of several species from the nearby Ashokan Reservoir. Mean richness across all sites decreased slightly during each year of the study, yet median richness was relatively consistent at 8.0, 7.5, and 8.0 species in 2009, 2010, and 2011, respectively, and did not differ between years.

Community diversity, estimated using 1 - D, generally reflected the same patterns observed for species richness (Fig. 2B). Diversity scores ranged from 0.15 at bush (DA = 29.5 km^2) in 2011 to 0.83 at esop6 (DA = 497.3 km^2) in 2011 (Table 2). Mean diversity scores for all 3 years were lower at all tributary sites (0.56) than

Table 2. Fish-community metrics from all sites surveyed in the Upper Esopus Creek during 2009–2011.

Site code	Richness (S)	Density (fish/0.1ha)	Biomass (g/0.1ha)	Index of diversity (1 - D)	Evenness (J')
2009 surveys					
fox	4	1707	13,517	0.35	0.48
peck	5	1396	9180	0.56	0.67
broad	7	881	4837	0.71	0.71
bush	4	2485	14,451	0.52	0.66
birch	7	1525	12,799	0.59	0.58
lbeav	9	657	4830	0.67	0.63
wood	7	2964	15,336	0.79	0.87
beav	11	1985	9891	0.72	0.65
stoc0	10	1790	10,749	0.74	0.66
stoc1	8	1440	7192	0.77	0.78
esop0	6	3474	9372	0.40	0.47
esop2	9	1517	40,843	0.79	0.80
esop3	8	2655	14,984	0.76	0.79
esop3a	10	2428	8307	0.80	0.79
esop3b	13	1260	3308	0.79	0.72
esop4	8	879	8743	0.72	0.72
esop4a	10	821	4696	0.81	0.79
esop6	15	1168	4993	0.69	0.62

they were at all downstream (0.62) and upstream (0.74) mainstem sites (Table 3). Like richness, mean diversity increased between tributary and mainstem sites, but unlike richness, the median values at upstream sites (0.76) did not differ significantly from median values at downstream sites (0.79). Furthermore, the average diversity at 3 sites (esop2, esop3, and esop3a) immediately upstream from the portal was comparable to the 4 downstream sites (Fig. 2B). Average and median species-diversity values at all Esopus sites were 0.59 and 0.72 in 2009, 0.67 and 0.70 in 2010, and 0.56 and 0.56 in 2011, respectively; medians did not differ significantly among years (Table 3).

Table 2, continued.

Site code	Richness (S)	Density (fish/0.1ha)	Biomass (g/0.1ha)	Index of diversity (1 - D)	Evenness (J')
2010 surveys					
fox	6	1567	147,06	0.66	0.68
peck	4	792	5638	0.57	0.77
broad	7	922	4248	0.77	0.81
bush	7	2375	9028	0.45	0.47
birch	9	1417	16,445	0.62	0.57
lbeav	10	1118	4943	0.77	0.73
wood	7	1215	7544	0.75	0.82
beav	9	2902	4533	0.69	0.63
stoc0	6	834	12,526	0.66	0.71
stoc1	7	817	3197	0.72	0.77
esop0	6	2219	11,032	0.22	0.30
esop2	9	1496	14,339	0.71	0.71
esop3	8	2436	11,205	0.70	0.69
esop3a	10	1301	5703	0.67	0.64
esop3b	11	533	1106	0.80	0.73
esop4	7	731	14,125	0.70	0.77
esop4a	9	2432	6854	0.79	0.77
esop6	15	850	2447	0.82	0.75
2011 surveys					
fox	4	516	3758	0.40	0.52
peck	5	703	4732	0.32	0.41
broad	6	622	4304	0.53	0.59
bush	7	3483	10,247	0.15	0.19
birch	6	1115	9554	0.53	0.60
lbeav	na	na	na	na	na
wood	7	497	2216	0.66	0.64
beav	10	2440	4643	0.50	0.45
stoc0	8	1029	4376	0.18	0.23
stoc1	8	438	2498	0.58	0.62
esop0	6	785	2801	0.48	0.48
esop2	8	1103	13,007	0.75	0.77
esop3	10	2570	9690	0.76	0.72
esop3a	11	1020	4222	0.76	0.73
esop3b	13	1072	2439	0.80	0.68
esop4	8	693	16,536	0.72	0.76
esop4a	na	na	na	na	na
esop6	13	680	3175	0.83	0.75

The spatial trend in density of fish communities was not as strongly related to DA as were richness and diversity (Fig. 2C). Total density ranged from 438 fish/0.1 ha at stoc1 in 2011 to 3483 fish/0.1 ha at bush in 2011 (Table 2). Mean density at all tributary sites (1503 fish/0.1 ha) differed little from that at upstream (1836
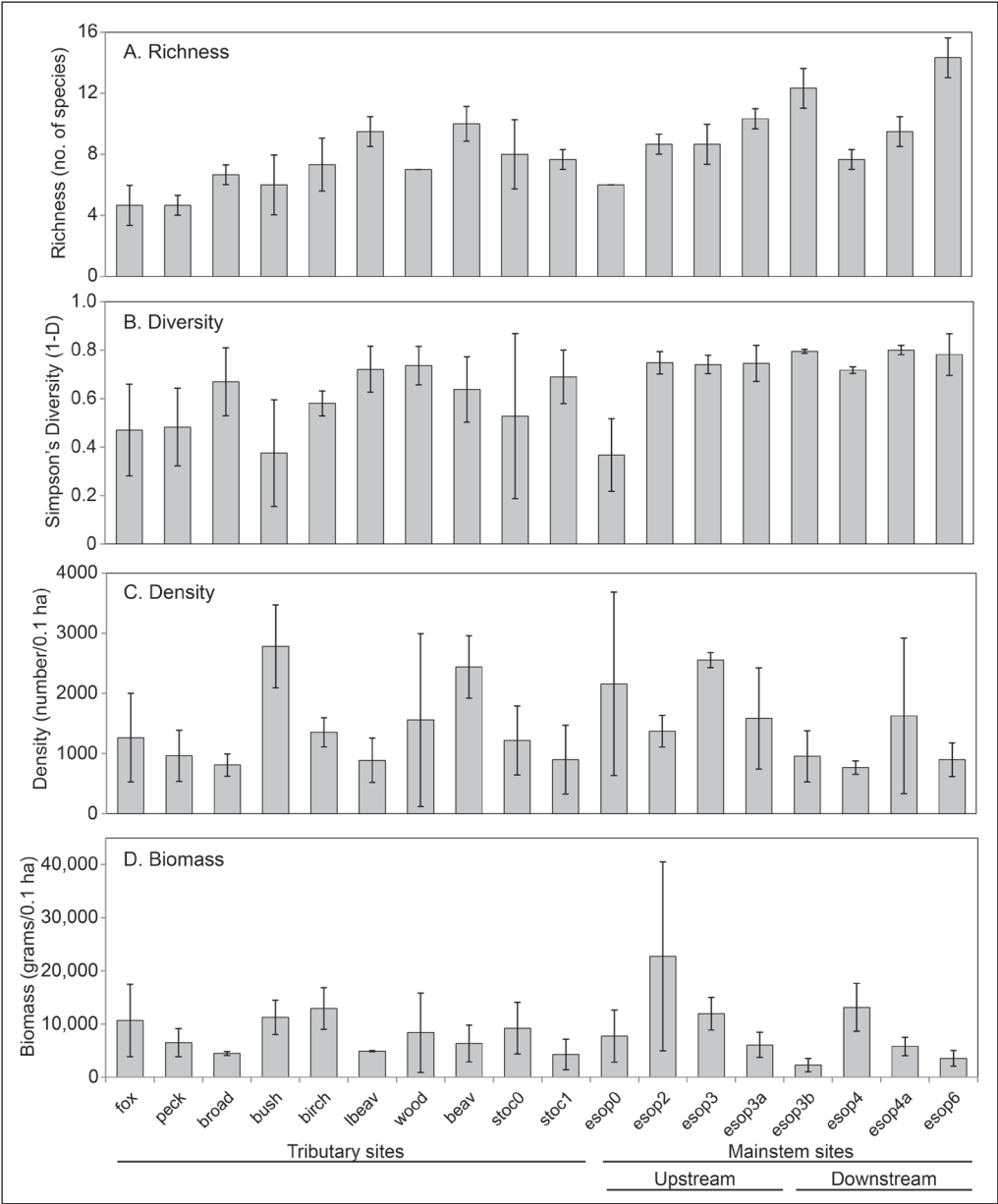

Figure 2. Mean richness (A), diversity (B), density (C), and biomass (D) and corresponding 95% confidence intervals (CIs) for resident-fish communities at all sites surveyed in the Upper Esopus Creek, 2009–2011. The locations of the sites (i.e., in a tributary and in the mainstem, either upstream or downstream from the portal) are denoted below the site identification codes; site esop0 functions as a tributary and a mainstem site for selected analyses.
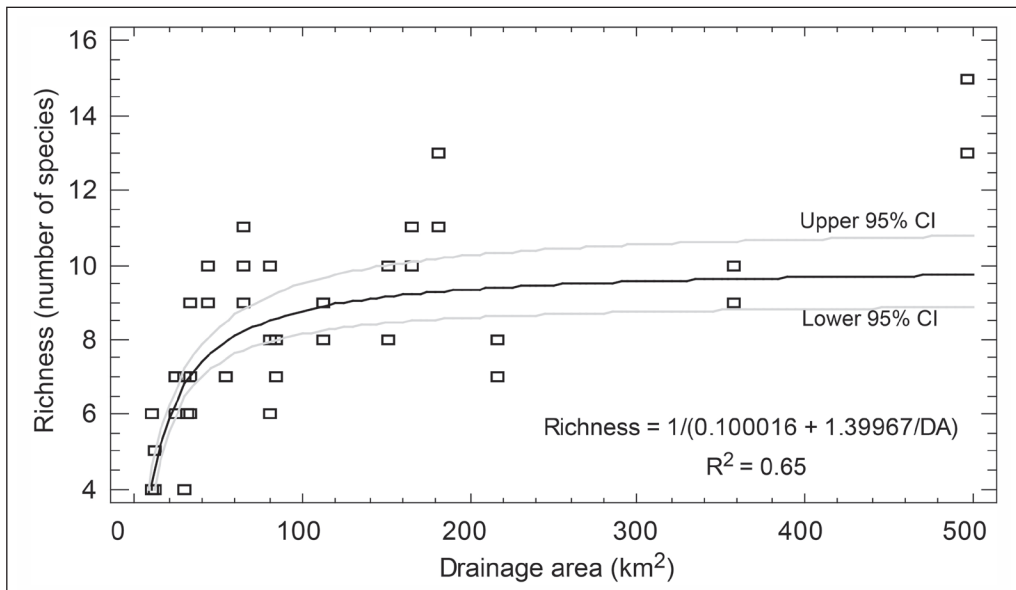
Figure 3. Relationship between community richness and drainage area, and 95% confidence intervals (CIs) for all sites surveyed in the Upper Esopus Creek, 2009–2011.

Table 3. Mean and median fish-community and trout-population metrics grouped by site type and survey year and the significance (*P*-values) for the differences in medians as determined by Kruskal-Wallis non-parametric tests. Significantly different group-medians are denoted by non-similar letters below each median. YOY = young-of-year, Trib = tributary sites, US = sites upstream from portal, DS = sites downstream from portal.

Metric	*P*-value	Site type			*P*-value	Survey year		
		Trib	US	DS		2009	2010	2011
Community richness								
mean		7.0	9.2	11.1		8.4	8.2	8.1
median	<0.001	7.0	9.0	11.0	0.9243	8.0	7.5	8.0
similar		a	b	c		a	a	a
Community diversity (*D*)								
mean		0.56	0.74	0.62		0.59	0.67	0.56
median	<0.001	0.59	0.76	0.79	0.3487	0.72	0.70	0.56
similar		a	b	b		a	a	a
Community density (fish/0.1 ha)								
mean		1503	1836	1011		1724	1442	1173
median	0.0313	1306	1517	850	0.0253	1521	1258	903
similar		ab	a	b		a	ab	b
Community biomass (g/0.1 ha)								
mean		7973	13,589	6220		11,002	8312	6137
median	0.0508	7368	11,205	4696	0.0310	9276	7199	4340
similar		a	b	a		a	ab	b
All trout density (fish/0.1 ha)								
mean		165.0	319.0	158.0		280.6	205.2	72.1
median	0.0697	138.6	357.7	148.9	0.0001	264.0	183.3	45.9
similar		a	b	a		a	a	b

Table 3, continued.

Metric	P-value	Site type			P-value	Survey year		
		Trib	US	DS		2009	2010	2011
All trout biomass (g/0.1 ha)								
mean		2621	3953	2326		3629	2926	1689
median	0.1791	2031	2804	3006	0.0037	3012	2031	1972
similar		a	a	a		a	ab	b
Brown Trout density (fish/0.1 ha)								
mean		91.1	101.7	24.9		137.7	65.9	27.6
median	0.0069	64.0	68.8	25.1	0.0001	138.0	54.6	21.3
similar		a	a	b		a	b	b
Brown Trout biomass (g/0.1 ha)								
mean		2124	2941	1863		3175	2314	1007
median	0.2395	1399	2747	857	0.0014	2742	1399	581
similar		a	a	a		a	ab	b
Mature Brown Trout density (fish/0.1 ha)								
mean		41.5	21.7	13.2		42.2	41.8	9.9
median	0.0112	36.4	22.2	11.8	0.0001	39.0	36.9	10.7
similar		a	ab	b		a	a	b
YOY Brown Trout density (fish/0.1 ha)								
mean		49.6	80.0	11.7		95.5	24.1	17.7
median	0.0157	20.3	33.9	9.3	0.0003	81.6	9.5	8.2
similar		ab	a	b		a	b	b
Percent YOY Brown Trout (%)								
mean		38.0	75.2	36.9		62.6	28.9	40.8
median	0.007	38.0	83.0	30.0	0.0053	65.9	31.2	34.7
similar		a	b	a		a	b	b
Rainbow Trout density (fish/0.1 ha)								
mean		71.3	213.3	132.5		140.4	135.8	43.1
median	0.0254	42.8	210.8	127.3	0.0075	114.9	127.8	27.5
similar		a	b	ab		a	a	b
Rainbow Trout biomass (g/0.1 ha)								
mean		475.7	975.0	461.9		432.5	595.2	661.1
median	0.5264	414.8	815.3	257.0	0.5769	372.3	507.8	376.7
similar		a	a	a		a	a	a
Mature Rainbow Trout density (fish/0.1 ha)								
mean		13.8	18.2	5.1		6.4	18.5	13.4
median	0.0309	12.2	16.6	2.0	0.0525	2.7	21.7	10.4
similar		ab	a	b		a	b	ab
YOY Rainbow Trout density (fish/0.1 ha)								
mean		57.5	195.2	127.4		134.0	117.3	29.8
median	0.0137	26.5	193.4	125.7	0.0030	105.3	111.8	13.9
similar		a	b	b		a	a	b
Percent YOY Rainbow Trout (%)								
mean		62.1	76.7	92.6		77.4	80.2	53.6
median	<0.001	71.4	92.1	15.1	0.0083	94.4	86.4	50.0
similar		a	ab	b		a	a	b

fish/0.1 ha) and downstream (1011 fish/0.1 ha) sites; however, the median density at downstream sites (850 fish/0.1 ha) was significantly lower than the median density at upstream sites (1517 fish/0.1 ha) (Table 3). Total community density declined at many sites (individually) and overall throughout the 3-year study. Mean and median density for all surveys at all sites ranged from 1724 and 1521 fish/0.1 ha in 2009, 1142 fish/0.1 ha and 1258 fish/0.1 ha in 2010, and 1173 fish/0.1 ha and 903 fish/0.1 ha in 2011, respectively; median density at all sites was significantly lower during 2011 than during 2009. Community density decreased from 2009 to 2011 in about 80% of the tributaries and half of the mainstem sites (see Fig. B in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). Based on 95% CIs, the 2011 estimates were significantly lower than in 2009 at most sites, and densities for 2010 were generally intermediate and not different from the 2009 estimates.

Community biomass (Fig. 2D), like density, was not strongly related to DA. Biomass ranged from 1106 g/0.1 ha at esop3b in 2010 to 40,843 g/0.1 ha at esop2 in 2009 (Table 2). Biomass averages and medians (g/0.1 ha) were, respectively: 7973 and 7368 in tributaries, 13,589 and 11,205 at upstream sites, and 6220 and 4696 at downstream sites. Estimates for median biomass at downstream sites were significantly lower than that for upstream sites (Table 3). Total biomass averaged about 9019 g/0.1 ha at sites esop3 and esop3a, immediately above the portal, and 7710 g/0.1 ha at sites esop3b and esop4, immediately below the portal. The lowest mean biomass for any site over most years was found at site esop3b, just downstream of the portal (Table 2; see also Fig. B in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). The low mean biomass (across all years) and the narrow CIs (±1255 g/0.1 ha) at this site (Fig. 2D) suggest that community biomass was significantly lower than at the next 2 upstream and next 2 downstream sites. Community biomass declined at many sites throughout the study period: it averaged 11,002 g/0.1 ha, 8312 g/0.1 ha, and 6137 g/0.1 ha for all surveys in 2009, 2010, and 2011, respectively. However, median biomass at upstream and downstream sites did not differ significantly (Table 3; see also Fig. B in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). Biomass estimates were lowest at 11 sites during 2011. Total biomass decreased from 2009 to 2011 in all of the tributaries, and in about 80% of the mainstem sites; these declines were significant between 2009 and 2011, and data for 2010 were generally intermediate or not different from 2009 or 2011.

Species populations

Changes in the distribution of several fish species accounted for most of the differences in community metrics across sites. Species distributions (richness) appeared to be influenced largely by stream size, which can be classified by DA. Small DA sites

(tributaries with DAs of less than 40 km^2) tended to be dominated by *Cottus cognatus* Richardson (Slimy Sculpin), trout, and *Rhinichthys cataractae* Valenciennes (Longnose Dace) (see Fig. C in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1), which are characteristic species in cold-water communities. Few other species contributed strongly to community density (see Table D in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1) or biomass at small DA sites (see Table E in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). Trout became more sporadically distributed and Slimy Sculpin was partially replaced by minnow and sucker species at tributary sites with moderately sized (40 km^2–100 km^2) DAs (see Fig. C in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). With minor exceptions, species density and biomass at most mainstem sites with DAs >100 km^2 generally decreased from upstream to downstream sites. Biomass of trout and Slimy Sculpin populations were very high at the 2 upper-most mainstem sites, esop2 and esop3, and declined at the sites immediately above and below the portal (at esop3a and esop3b). Seven species—*Luxilus cornutus* (Mitchill) (Common Shiner), *Etheostoma olmstedi* Storer (Tessellated Darter), *Noturus insignis* (Richardson) (Margined Madtom), *Notemigonus crysoleucas* Mitchill (Golden Shiner), *Lepomis cyanellus* Rafinesque (Green Sunfish), *Micropterus salmoides* Lacépède (Largemouth Bass), and *Alosa pseudoharengus* (Wilson) (Alewife)—were first encountered at these 2 sites, i.e., they were absent from all mainstem sites upstream of esop3a (see Table D in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). Species populations at esop4 also did not reflect the expected community composition given its drainage area. Trout and Slimy Sculpin density and biomass increased markedly and density and biomass of minnow species declined at esop4 relative to the 2 prior upstream sites (see Fig. C in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1). Increases in water velocity and decreases in temperature (related to inputs from the portal) may be partially responsible for observed community changes. Species diversity and richness increased and additional centrarchids, as well as other minnow species, were present at the 2 lower mainstem sites, esop4a and esop6. Four species—*Micropterus dolomieu* Lacépède (Smallmouth Bass), *Ambloplites rupestris* (Rafinesque) (Rock Bass), *Semotilus corporalis* Mitchill (Fallfish), and *Pimephales notatus* (Raphinesque) (Bluntnose Minnow)—were only collected at esop6.

Trout populations

Understanding spatial and temporal patterns in trout populations was a primary focus of this study. Spatial trends in total density and biomass of trout populations

at all tributary and mainstem sites were not generally predictable based on DA (Fig. 4). In fact, DA only explained 2–3% of the variability in the density or biomass of all trout (pooled) populations at all sites in the basin. Trout density averaged 200 fish/0.1 ha at the 6 small tributary sites (DA < 40 km^2), 120 fish/0.1 ha at the 5 large tributary (DA = 40 km^2–100 km^2), and 230 fish/0.1 ha at the 7 mainstem (DA > 100 km^2) sites (excluding esop0) (Fig. 4A). Trout biomass averaged 3015, 2113, and 3058 g/0.1 ha at the small tributary, large tributary, and mainstem sites, respectively (Fig. 4B). Median trout density was significantly lower at downstream sites (149 fish/0.1 ha) than at upstream sites (358 fish/0.1 ha) (Table 3). Median trout biomass, however, did not differ significantly between mainstem sites located upstream and downstream from the portal.

We collected Brown Trout and Rainbow Trout at most sites. Estimates of total biomass for both species collected from all sites each year during 2009–2011 are summarized in Figure 5. The average density of Rainbow Trout populations at all sites (169 fish/0.1 ha) was about 3-fold higher than that of Brown Trout (59 fish/0.1 ha), however average biomass of Brown Trout populations at all sites (2347 g/0.1 ha) was about 3-fold greater than that of Rainbow Trout populations (693 g/0.1 ha) (Table 3).

The portal may affect Brown Trout and Rainbow Trout populations indirectly by altering survival of their early life stages (e.g., eggs, swim-up fry, and young-of-year), thus altering the proportions of juvenile and mature fish. We used non-parametric Kruskal-Wallis tests to assess differences in median density and biomass for both species and for differences in median densities and percentage of young-
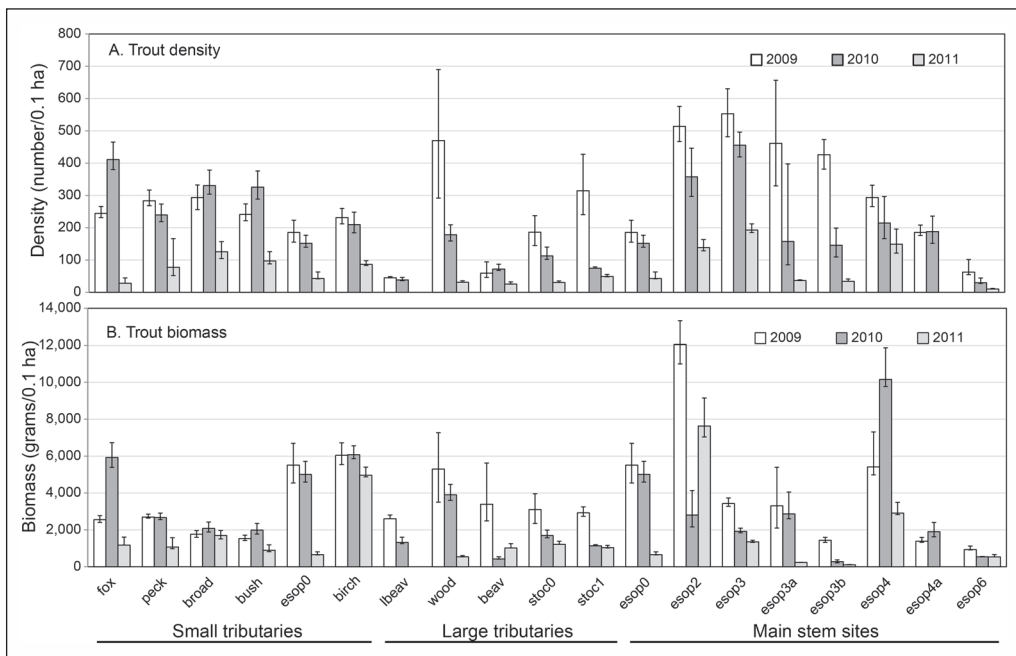

Figure 4. Estimates of total density (A) and biomass (B) and 95% confidence intervals (CIs) from all trout at sites surveyed in the Upper Esopus Creek during 2009, 2010, and 2011.

of-year (YOY) Brown Trout and Rainbow Trout at 3 upstream, 4 downstream, and 11 tributary sites (Table 3). The median density of Brown Trout populations was significantly lower at downstream sites (25 fish/0.1 ha) than at upstream sites (69 fish/0.1 ha), but median biomass did not differ significantly among the 3 site types. Median densities of mature Brown Trout did not differ significantly between downstream and upstream sites. Median density of YOY Brown Trout was significantly lower at downstream sites (9 fish/0.1 ha) than upstream sites (34 fish/0.1 ha). The YOY constituted 38%, 75%, and 37% of Brown Trout populations at tributary, upstream, and downstream sites, respectively, and the median percentages were significantly lower at downstream sites (30%) than at upstream sites (83%). Median density and biomass estimates for Rainbow Trout populations did not differ significantly between downstream and upstream sites (Table 3). Median density of mature Rainbow Trout was significantly lower at downstream sites (2 fish/0.1 ha) than at upstream sites (17 fish/0.1 ha). Median densities of YOY Rainbow Trout did not differ significantly between downstream and upstream sites. The YOY constituted 62%, 77%, and 93% of Rainbow Trout populations at tributary, upstream, and downstream sites, respectively, and median percentages did not differ significantly between downstream and upstream sites.

Temporal and spatial variability in fish assemblages

The temporal trends in fish communities and trout populations were relatively similar. Community density and biomass decreased significantly at many sites (e.g., stoc1, fox, and esop3a) (see Fig. B in Supplemental File 1, available online at http://www.eaglehill.us/NENAonline/suppl-files/n22-2-N1280-Baldigo-s1, and for BioOne subscribers, at http://dx.doi.org/10.1656/N1280.s1), and overall at all sites
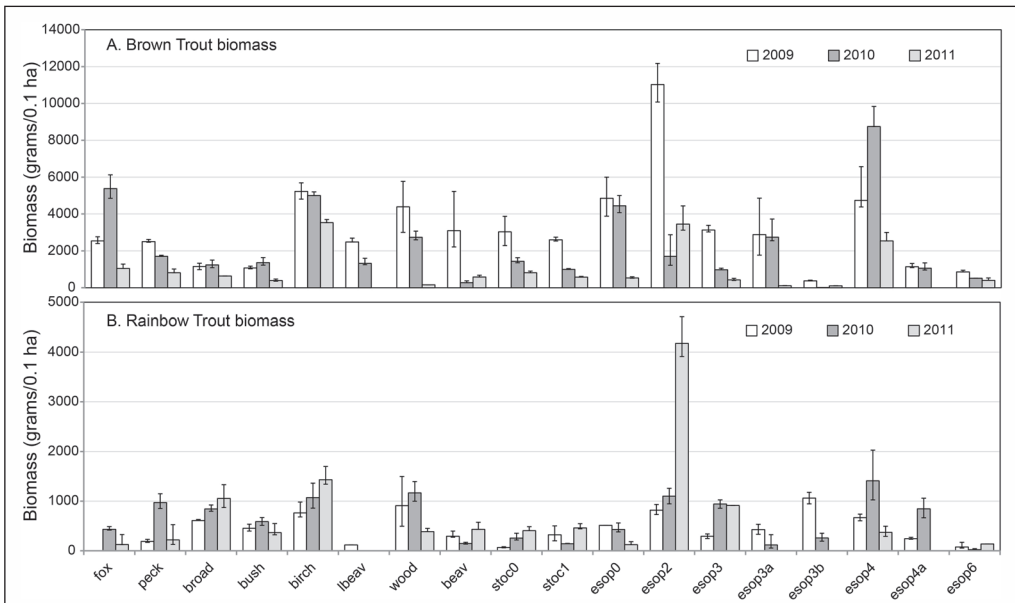

Figure 5. Estimates of Brown Trout biomass (A) and Rainbow Trout biomass (B) and 95% confidence intervals (CIs) at sites surveyed in the Upper Esopus Creek during 2009–2011.

(Table 3) between 2009 and 2011. Temporal trends in density and, to some degree, biomass of trout populations were also significant and relatively consistent at many sites (Table 3, Fig. 4). The density of trout populations at most sites was generally highest in 2009, lower in 2010, and lowest in 2011; many of these differences were significant between years (Fig. 4A). The changes in trout biomass over the 3 periods were comparable to density changes at several sites (e.g., esop3, wood, and stoc0), yet the observed decreases were less regular, often not significant, and sometimes interrupted by increases during 2010 (Fig. 4B). The few exceptions to these temporal trends were generally related to collection of a few large trout, which strongly influenced biomass estimates at individual sites such as esop2 during 2011 and esop4 during 2010. Median density of all trout was significantly lower in 2011 (46 fish/0.1 ha) than in 2009 (264 fish/0.1 ha) and 2010 (183 fish/0.1 ha), and median biomass was significantly lower in 2011 (1972 g/0.1 ha) and 2010 (2031g/0.1 ha) than in 2009 (3012 g/0.1 ha) (Table 3). In general, all mean and median population metrics for Brown Trout decreased significantly from 2009 to 2011; most decreases were also significant between 2009 and 2010 (Fig. 5A). Three median Rainbow Trout population metrics decreased significantly between 2009 and 2011, but only the decrease in density of mature Rainbow Trout was significant between 2009 and 2010 (Fig. 5B).

Multidimensional scaling ordinations and cluster analysis using species-density data identified 5 unique site groupings distributed roughly along a stream-size or drainage-area gradient (Fig. 6). The composition of fish assemblages at all sites (and survey years) within each group was 58–72% similar to each other and differed significantly ($P < 0.05$) among groups. The groups consisted of (1) only small headwater tributaries, which were 67% similar; (2) large tributaries, Broadstreet Hollow (broad), and mainstem sites mostly upstream of esop4a, which were 53% similar; (3) large tributaries and one upstream site (esop3a), which were 58% similar; (4) esop6 during 2010 and 2011, which were 64% similar; and (5) esop6 and esop3a during 2009, which were 72% similar. The last 2 groups were also 55% similar to each other (not shown) and are treated as 1 group in the discussion. The overall groupings confirm major differences among fish assemblages observed at small tributaries, the largest downstream site, and all other sites, yet they identify no consistent differences between fish assemblages at sites upstream and downstream of the portal. Density bubble-plots for Slimy Sculpin and *Exoglossum maxillingua* Lesueur (Cutlips Minnow) illustrated pronounced trends along the horizontal axis (Fig. 7). Slimy Sculpin populations were generally most dense at small headwater tributaries (1800–3100 fish/0.1 ha at bush) and essentially absent at large downstream mainstem sites during all years (Fig. 7A), whereas Cutlips Minnow populations were largest (600–800 fish/0.1 ha) at upstream (esop3a) and downstream (esop6) mainstem sites during 2009 (Fig. 7B). A SIMPER analysis confirmed that Slimy Sculpin was responsible for as much as 32.4% of the dissimilarity among small tributary, large tributary, and mainstem groups. Cutlips Minnow was almost as influential as Slimy Sculpin and contributed as much as 19.6% to the dissimilarities among groups. These findings indicate that Slimy Sculpin and Cutlips Minnow may be suitable indicator species for classifying community types in this basin.

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Discussion

In general, the results from this investigation indicate that fish assemblages in the Esopus respond to waters from the portal in a variety of ways. The most important finding was that no adverse changes in fish assemblages could be attributed directly to portal waters, and some of the noted effects might even be considered beneficial. Though fish communities were altered at sites near and downstream from the portal, many changes were positive and could be related to changes in temperature or habitat quality and quantity. One companion study of fatty acids and periphyton communities in the upper basin (S.D. George, US Geological Survey, Troy, NY, unpubl. data) detected the only ecological impairment that may be linked directly to the portal: a reduction in the standing crop of primary producers at sites immediately downstream from the portal. Potentially adverse effects were also noted in the density and biomass of juvenile life stages of Brown Trout. Though such effects could be caused by impaired water quality, the portal was not a major contributor to turbidity and suspended sediment loads in the up-
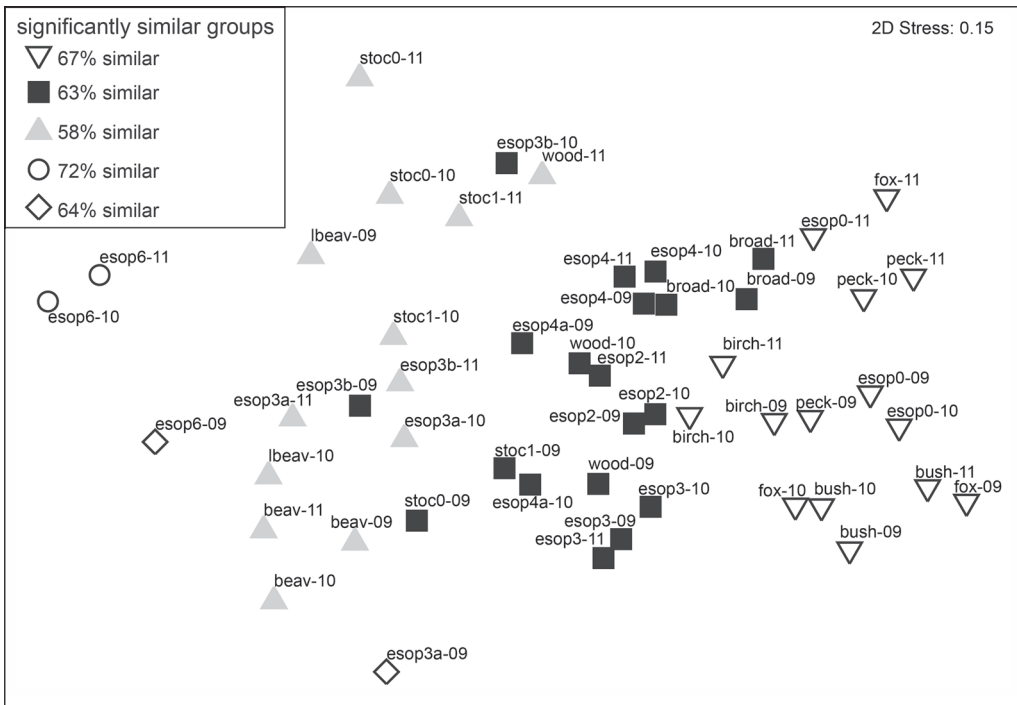

Figure 6. Non-metric MDS ordination plot of fish assemblages based on square-root-transformed density of all species collected at 16–18 sites in the Upper Esopus Creek annually from 2009–2011. The 5 symbols denote site membership in groups with significantly similar communities (58–72% Bray-Curtis similarity, $P \leq 0.05$) based on group-averaged cluster analysis. Site locations are shown in Figure 1.

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Figure 7 (following page). Density-bubble plots showing the distribution and density of (A) Slimy Sculpin and (B) Cutlips Minnow populations across sites and years (labels on bubbles) in the Upper Esopus Creek, 2009–2011. Site locations are shown in Figure 1.

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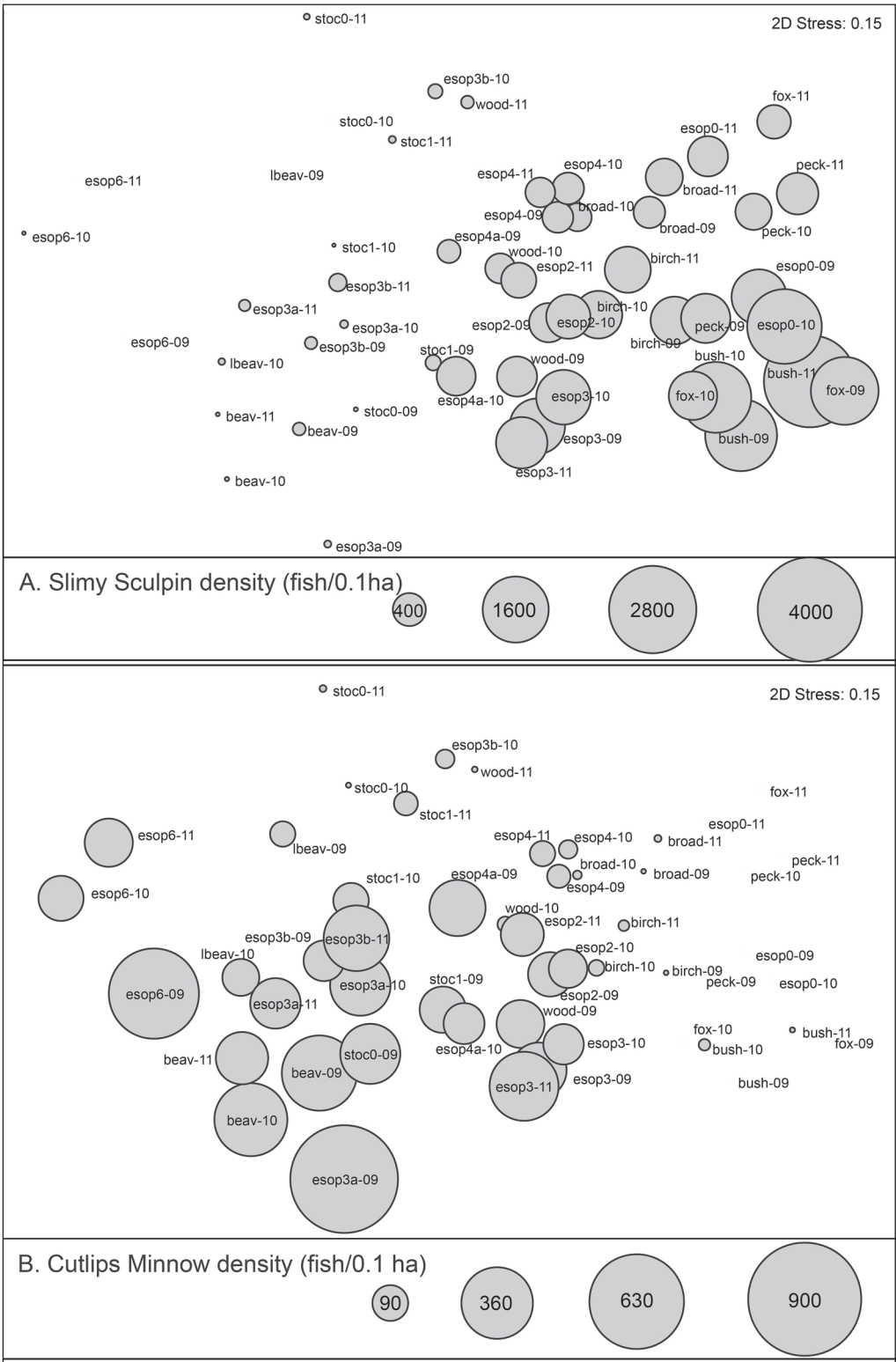
per basin (McHale and Siemion 2014). Loads typically reflect short-duration (high-flow) events and, thus, may not be as important to trout growth and survival as long-term exposure to moderately elevated levels of suspended sediment and turbidity. During water-years 2010 and 2011, median turbidity from 34 routine grab samples collected on the same dates (every 2–4 weeks) upstream from the portal at esop03a (7.5 NTRUs) and downstream from the portal at esop03b (9.4 NTRUs) (USGS 2014) did not differ significantly (Mann-Whitney U-Test: $P =$ 0.1990), nor did the distributions of turbidity data from both sites (Kolmogorov-Smirnov Test: $P = 0.1601$). The median turbidity for 24 mean-daily values (estimated monthly from continuous data) at the portal during the same 2-year period was 16.4 NTU (EPA 2014). Although these results indicate that waters from the portal slightly reduced clarity at downstream sites, turbidity levels generally remained below thresholds (approximately 40 NTU) found to impair feeding, growth, and survival of several trout species (Newcombe 2003, Rowe et al. 2003, Stuart-Smith et al. 2004, Sweka and Hartman 2001b), which suggests that these small increases may not be biologically meaningful.

When compared to richness at other mainstem sites, the significantly larger number of species at sites esop3a and esop3b, immediately upstream and downstream of the portal, suggests that the portal's connection to the Schoharie Basin was the source of several fish species. Similarly, richness was high at the furthest downstream site (espo6), near the Ashokan Reservoir where we encountered as many as 15 species. The richness of fish species at sites in the nearby Beaverkill Basin, with the same drainage area as esop6, ranged from 12 to 14, whereas comparable sites in the adjacent Neversink Basin (which are close to its confluence with the Delaware River) reach as high as 18 species (Baldigo, et al., in press). Richness at sites in the Beaverkill with the same drainage area as esop3a and esop3b ranged from 10 to12 species, whereas 7 species were collected at the 1 comparable site in the Neversink, which was upstream of a reservoir. Although the small increases in richness at sites near the portal (and at esop6) did not cause significant differences in diversity, the addition of species creates more complex food webs, which could benefit local communities by making ecosystems less susceptible to the effects of short- and long-term stresses or perturbation (Schaefer et al. 2012). The increased richness at esop6 was likely related to its close proximity to the reservoir and to its habitat, which is more suitable for these lentic species than sites further upstream. Although increased species richness may be beneficial, additional nonnative species could also be detrimental to resident fish communities (Cucherousset and Olden 2011).

Waters from the portal had a number of measureable effects on fish communities at downstream mainstem sites that suggest the normal upstream–downstream continuum was disrupted. Estimates of total density and biomass for fish communities at individual mainstem sites revealed no significant differences that could be directly attributed to the portal, yet total density and biomass at 4 mainstem sites downstream were significantly lower than at the 3 mainstem sites upstream. Conversely, the multivariate analysis detected no significant impacts of the portal

on fish communities at mainstem sites downstream. More important, however, may be the increased number of fish species at several sites (e.g., esop3a and esop3b) in the Esopus. Fish assemblages (i.e., the distribution of individual species) in large undisturbed basins normally follow a predictable succession (longitudinal zonation) between small, cold, low-order headwater reaches, and large, warm, high-order reaches reflecting a continuum of abiotic and biotic factors (Vannote et al. 1980). Fish-species richness and diversity normally increase progressively with increasing stream size and order (at low to mid-order reaches), often with additions to, rather than replacement of, existing species (Hutchinson 1993, Whiteside and McNatt 1972, Zalewski et al. 1990). In general, physical and chemical factors including dissolved oxygen, pH, alkalinity, suspended sediment turbidity, and conductivity often change with stream size; however, stream depth, size, order, and habitat heterogeneity typically explain the largest amount of variability in richness and diversity of fish communities (Hutchinson 1993; Schlosser 1987, 1991). In temperate rivers, abiotic factors can limit the diversity of low-order fish communities to relatively few species that are tolerant of lower temperatures, wide fluctuations in flow, small/shallow channels with homogeneous habitat, and high water-velocities. High rates of primary production in mid-order stream reaches can increase species richness, biodiversity, density, and/or levels of biomass for secondary and tertiary consumers (Ward and Stanford 1983). Other biotic factors, such as competition and predation, typically control richness and diversity of fish communities at larger downstream reaches where relatively stable flows, more heterogeneous stream channels and habitat, and slower and warmer waters permit additional species to coexist (Vannote et al. 1980). Mid-basin changes in richness and diversity indicate a small and localized effect on the Esopus ecosystem, yet decreases in fish-community biomass and density indicate that primary production, along with secondary and top consumers, may have been unusually low at several sites downstream from the portal.

Altered composition of periphyton communities could be responsible for decreases in primary production at mid-order sites in the Esopus. Periphyton is responsible for most primary production and forms the base of the food web in low- to mid-order streams in temperate climates (Vannote et al. 1980). Chlorophyll *a* (chl *a*) concentration and ash-free dry mass (AFDM) quantify the standing crop of primary producers (assumed herein to be a surrogate for primary production) in stream food webs (Rosemond et al. 1993). Although neither chl *a* nor AFDM differed significantly between groups of upstream and downstream sites, these values were significantly depressed at the first 3 downstream sites (esop3b, esop4, and esop4a) compared to the 3 upstream sites (Smith et al. 2013). Large decreases in primary production can cause cascading effects at higher trophic levels (e.g., primary consumers, predators; Kurle and Cardinale 2011). Changes in the abundance and/or biomass of certain preferred (or all) macroinvertebrate-prey species could theoretically limit available food resources, and therefore, growth and survival of some fish species, at one or more downstream sites. The associations between standing crop, periphyton communities, and macroinvertebrate communities are

relatively strong in the Esopus (Smith et al. 2013) and provide additional evidence that the food web is altered at several downstream sites. Because changes in primary production can cause cascading effects throughout the food web (Henley et al. 2000, Vannote et al. 1980), direct and indirect impacts to predator (fish and macroinvertebrate) populations are possible.

The indirect effects of reduced primary production and the direct effects of unusual thermal, hydrologic, and water-quality regimes on the 2 dominant trout species are potentially important, but difficult to isolate in this system because of their dissimilar life-histories. Although both types of effects may be attributed to the portal, the disparate timing for spawning, incubation, and emergence (from gravel beds) make the early life stages for each species differentially vulnerable to storm flows occurring at different times of the year. Nevertheless, the median density of trout at downstream sites was significantly lower than that at sites upstream from the portal, and biomass differences were only nominal. Differences in median Rainbow Trout metrics were generally minor; thus, changes in Brown Trout populations drove differences in overall trout metrics. Median densities of all Brown Trout and YOY Brown Trout were significantly lower at downstream sites than at upstream sites, yet densities of mature Brown Trout did not differ between the 2 reaches. Ross (2012) determined that upstream reaches contained more habitat preferred by juvenile Brown Trout than did downstream reaches, which agrees with the significant differences we detected in densities of YOY Brown Trout between sites from both reaches. Although the 37% decrease in biomass of Brown Trout populations between upstream and downstream sites was not significant, the significantly lower density of YOY Brown Trout suggests that their populations might have been larger if not for their location downstream of the portal. Clearly, the abundance of juvenile Brown Trout was reduced at many downstream Esopus sites.

Though the potential effects of the portal on trout provided the impetus for this investigation, a number of related mechanisms could also adversely affect productivity and fish assemblages downstream. This consideration is important because observational studies can only describe the strength of the relationships between or among factors; the findings cannot unequivocally attribute a specific effect to a specific cause. Assuming that turbid waters directly reduce primary productivity, then the portal would only play a nominal role because it has been a minor source of turbidity and suspended sediments since the mid-1990s (CCE 2007). For example, estimates from all samples collected during water-year 2010 for median turbidity (NTRU) varied from 119 at Stony Clove, 26 at Broadstreet Hollow, 41 at Woodland Valley Creek, and 47 at Beaverkill, to 77 at the last mainstem site (esop3a) upstream of the portal, and 20 at the first site (esop3b) downstream of the portal (McHale and Siemion 2014, USGS 2014). The median turbidity for 12 mean-daily values (estimated monthly from continuous data) reported for waters discharged from the portal during water-year 2010 was 15.3 NTU (EPA 2014), which suggests that portal waters do not substantially alter natural turbidity levels within the Esopus. Thus, turbidity from portal waters probably does not limit fish-species distributions and their assemblages at downstream Esopus sites. These data

indicate that several tributaries, not the portal, are the primary sources of turbidity in the upper basin. In fact, Stony Clove Creek accounted for more of the total suspended sediment load (30–57%) at the furthest downstream site, esop6, during water-years 2010 and 2012 than did all other tributaries combined (McHale and Siemion 2014).

The results from other investigations show that elevated turbidity levels can strongly affect the behavior, growth, and condition of Brown Trout and other salmonids, yet few have detected strong linkages in the wild (Henley et al. 2000). During laboratory experiments, significant changes in behavior and activity levels of Brook Trout only occurred when turbidity levels surpassed 7.1 NTU (Gradall and Swenson 1982). Increasing turbidity levels from 0 to 45 NTU had no significant effect on the rates at which Brook Trout consumed prey because they switched from passive to more active searching and feeding behavior (Sweka and Hartman 2001a). Although growth rates of Brook Trout decreased linearly with increasing turbidity, they did not differ significantly from controls until turbidity levels reached 45 NTU, when they declined by 62% (Sweka and Hartman 2001a). Similarly, the volume of stomach contents and prey diversity for Brown Trout sampled from Lake Sorell, Tasmania, were greater in 1996 when turbidity averaged 26 NTU than they were during 2001 when turbidity averaged 141 NTU (Stuart-Smith et al. 2004). Repeated exposure to 0–6-d pulses of turbid (suspended sediment concentrations of 700 mg/L) water also significantly affected growth of juvenile Rainbow Trout in streamside (caged) experiments (Shaw and Richardson 2001). Model simulations predicted that if Rainbow Trout were exposed to high turbidity (53 NTU), they would occupy shallower- and slower-water habitats, switch from passive to active prey-capture behavior, gain no net energy, and remain in poor condition (Harvey and Railsback 2009). Despite these findings, Rainbow Trout had normal feeding rates (White and Harvey 2007) and wild trout populations generally persisted (Harvey and Railsback 2009) in streams with highly turbid regimes, purportedly because trout can sense and capture prey using non-visual organs (e.g., the lateral line) under highly turbid and low-light conditions (Rowe et al. 2003).

The supplemental flows from the portal could potentially benefit the health of resident trout and their populations at downstream Esopus sites in several ways. The intakes for the Shandaken tunnel are deep in the Schoharie Reservoir and withdrawals usually originate in the hypolimnion. Thus, water temperatures for these enhanced flows are typically lower at downstream Esopus sites than they are at upstream sites during the warm months. The positive effect of the supplemental flows on habitat quantity (area and volume) in the upper basin is illustrated by adding flows at esop3a, upstream of the portal (USGS 2012a), which averaged 97 ft^{3}s^{-1}, to those from the portal (USGS 2012b), which averaged 226 ft^{3}s^{-1} during summer base-flow periods (July–September) between 1996 and 2011. Simple addition shows that Esopus flows immediately downstream from the portal averaged at least 323 ft^{3}s^{-1} during these same months, and would have been about 77% lower without supplemental flows from the Schoharie Reservoir.

During the warmest months of June, July, and August 2011, water temperatures averaged 11.4, 14.0, and 17.2 °C, respectively at the portal; 14.9, 17.9, and 17.8 °C, respectively at the upstream site esop3a; and 12.8, 15.6, and 17.3 °C, respectively at the downstream site esop3b. Except for August 2010 (when the Schoharie Reservoir reached unusually low levels and epilimnion withdrawals replaced hypolimnion withdrawals), water temperatures during June–August in 2009–2011 averaged 2.7 °C lower at the portal than at esop3a, and temperatures averaged 1.8 °C lower at esop3b than at esop3a. The large volumes of cold water from the portal have an important effect on summer stream temperatures at Esopus reaches downstream from the portal. Such thermal differences are crucial to the viability of Brown Trout populations in reaches where water temperatures approach and sometimes surpass thermal limits for growth and survival (Elliott and Elliott 2010, Wehrly et al. 2007).

The findings by Ross (2012) help explain our conflicting results that identified substantial effects on the densities of juveniles (YOY), yet only minor effects on densities of mature Brown Trout at downstream sites. Ross (2012) reported that biomarkers (serum chemistry, gill histology, and water content) and growth rates indicated that Brown Trout were generally stressed at all reaches (upstream, downstream, and further downstream), but that the portal had no significant impact on the species at downstream reaches in the Esopus. In general, Ross (2012) determined that (a) the health and growth rates of adult Brown Trout were poor across the Upper Esopus, (b) the quality of habitat for adult trout was optimal at downstream sites, and (c) the quality of habitat for juvenile trout was poor at downstream sites. These findings partly explain why mature Brown Trout densities were generally unaffected and YOY Brown Trout were less abundant at downstream sites. Possibly more notable was the fact that growth of Brown Trout, during warm months, was less negative in reaches immediately downstream from the portal than growth of trout from upstream and further downstream reaches. The inference is that conditions downstream from the portal were less stressful than conditions at upstream and further downstream reaches during the warm months. These data suggest that a stress-refuge may exist periodically at sites immediately downstream from the portal, yet our results indicate that no broad benefit was conveyed to resident Brown Trout or Rainbow Trout populations at downstream sites.

The results from annual surveys generated baseline information for an important natural resource and also detected decreasing temporal trends in many fish-community and population metrics. These data are important because they define the current status of local fisheries and quantify the normal measures of error (natural variability) that may occur in key metrics given the typical year-to-year variations in temperature, precipitation, and stream flows. As such, these results provide a baseline dataset that defines the fishery status during the period 2009–2011. More important, these data are now available to quantify the impacts of short-term (e.g., severe flooding or contaminant spills) or long-term (e.g., altered thermal or hydrologic regimes associated with climate change) disturbances on resident-fish assemblages. The decline in many metrics between 2009 and 2011 may reflect natural responses to a

drought during summer 2010 and moderate floods during the 2011 water-year (before we conducted our fish surveys). Although speculative, low flows and warmer than normal temperatures during summer 2010 could have caused widespread stress and mortality among resident-fish species, and flood flows during fall 2010 and spring 2011 could have negatively affected the survival of eggs and YOY Brown Trout that spawned in the fall or Rainbow Trout that spawned in the spring. Thus, depressed fish populations during 2010 could have further adversely affected the density and biomass of species populations during 2011. Regardless of the reasons, site-to-site and year-to-year variations in most fishery metrics are much larger and more notable than any effects that the portal may have had on fish populations.

In summary, from 2009–2011, the fish assemblages at most sites in the Esopus were comparable to those previously observed in other rivers of the region. Community richness increased near the portal, and median densities and biomass of fish communities generally declined significantly at the downstream sites. Such effects, however, do not suggest a specific cause and could easily be related to a general shift to larger and fewer individuals, some sampling bias, differences in sampling efficiency, changes in habitat, or poor water quality at sites downstream from the portal. A decrease in water quality would implicate tributaries because they were the primary sources of increased turbidity and suspended sediments loads at all mainstem reaches downstream of the portal. The only obvious deleterious effect of the portal seemed to be a large decrease in the standing crop of primary producers as indicated by changes in chl *a* and AFDM at sites immediately downstream from the portal (Smith et al. 2013). Less conspicuous, the densities of YOY Brown Trout were significantly reduced at several downstream sites, but associated impacts are not transferred to their whole populations. The increased water volume and decreased temperature at downstream reaches during summer months also increase physical habitat and help moderate temperatures that would otherwise be harmful to resident trout. Although such conditions may not truly be beneficial, the conditions at mainstem sites immediately downstream from the portal are clearly less harmful to trout (and their populations) than are conditions at upstream and further downstream sites during warm months. Despite the notable decreases in primary production and density of YOY Brown Trout at several sites, this study shows that fish communities and trout populations across most reaches in the Esopus are generally in good condition and unaffected by the portal.

Acknowledgments

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